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Simulating stream response to floodplain connectivity and revegetation from reach to watershed scales: Implications for stream management

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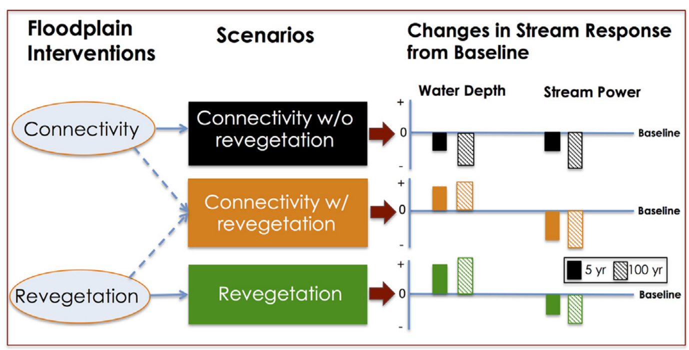
HIGHLIGHTS

- Influence of floodplain connectivity and revegetation on ecosystem services provided by stream was quantified
- Interventions influenced different aspects of stream response in diverse ways and varied widely along reaches
- Landscape and morphology of reaches may determine the effectiveness of interventions
- Individual interventions have their own benefits and shortcomings between target and non-targeted areas
- Careful evaluation is needed to compare benefits and costs among interventions

GRAPHICAL ABSTRACT



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ABSTRACT

Natural-infrastructures (e.g., floodplains) can offer multiple ecosystem services (ES), including flood-resilience and water quality improvement. In order to maintain these ES, state and non-profit organizations consider various stream interventions, including increased floodplain connectivity and revegetation. However, the effect of these interventions is rarely quantified. We build a hydraulic model to simulate the influence of above-mentioned interventions on stream power and water depth during 5 yr and 100 yr flood return-intervals for two watersheds in Vermont, USA. Simulated revegetation of floodplains increased water depth and decreased stream power, whereas increasing connectivity resulted in decline of both responses. Combination of increased connectivity and floodplain revegetation showed greatest reduction in stream-power suggesting that interventions may influence stream response in diverse ways. Across all three interventions, 14% and 48% of altered reaches showed increase in stream power and water depth over baseline, indicating that interventions may lead to undesirable outcomes and their apparent effectiveness can vary with the measure chosen for evaluation. Interventions also influenced up to 16% of unaltered reaches (i.e., in which no interventions were implemented), indicating the consequences of interventions can spread both up and downstream. Multivariate analysis showed that up to 50% of variance in stream response to interventions is attributable to characteristics of reaches, indicating that these characteristics could mediate the effectiveness of interventions. This study offers a framework to evaluate the potential ES provided by natural infrastructure. All stream interventions involve tradeoffs among responses and between target and non-target areas, so careful evaluation is therefore needed to compare benefits and costs among interventions. Such assessments can lead to more effective management of stream-floodplain ecosystems both in Vermont and elsewhere.

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1. Introduction

Stream-floodplain ecosystems (hereafter stream-floodplain) are some of the most productive on the earth (Tockner and Stanford, 2002) and offer a range of ecosystem services (ES), including supporting unique habitat and biodiversity, improving water quality, reducing flooding and providing recreation value (Costanza et al., 1997; Brauman et al., 2014; Hanna et al., 2017). Simultaneously, stream-floodplain systems are also one of the most threatened and heavily managed, mostly to further exploit the services provided by these unique ecosystems (Tockner et al., 2010; Bhattacharya et al., 2016). Subsequently, these activities have resulted in modification of >95% of streams in the Northern Hemisphere (Vitousek et al., 1997).

To address the continuing degradation of stream-floodplains over the past several decades, restoration managers have been using geomorphic form and structure based interventions, such as straightening of streams, altering flow patterns via flow deflectors and boulders and removing dams and levees to minimize ecological impact (c.f., Bernhardt et al., 2005). Over time these restoration practices have become a billion-dollar enterprise in the United States (Bernhardt et al., 2005) and are also widely used worldwide (Arthington and Pusey, 2003; Nakamura et al., 2006; Jeong et al., 2011; Rinaldi et al., 2011). Most of these restoration activities are done under the assumption of restoring streams to the pre-disturbance condition or a certain reference that is rarely known (Palmer et al., 2005, 2014a). Due to limitations of restoring streams to an unknown reference, restoration ecologists have been looking at other alternatives that can integrate river function and processes with socioeconomic benefits that rivers may provide (Dufour and Piegay, 2009). In order to attain these objectives, restoration ecologists have started to support nature-based solutions that can simply maintain ecosystem services provided by these floodplains (Costanza et al., 1997), while preserving the overall health of streams (Gilvear et al., 2013; Palmer et al., 2014b; Hanna et al., 2017).

Conservation organizations, like The Nature Conservancy, have also incorporated these nature-based solutions into their conservation activities to maintain healthy habitat for biodiversity and to minimize flooding and water quality issues downstream (TNC, 2017). Nature-based solutions are also being implemented in streams and floodplains across the European Union to increase their resilience to flooding (Baptist et al., 2004; Leyer et al., 2012). Given the rise in recognition of nature-based solutions for increasing flood protection and mitigating water quality issues, there is a need to understand how these nature based solutions affect ecosystem services provided by floodplains. In particular, the influence of nature-based solutions on hydrologic and geomorphic behavior of floodplains in ways that affect their ES remains unclear.

Many natural floodplain-based interventions include revegetation and increasing stream-floodplain connectivity. Revegetating floodplains has played a role in many restoration activities focusing on stabilizing stream banks, reducing sediment and nutrient loads, and mitigating flooding (Bernhardt et al., 2005). Studies have shown how vegetation biomechanics (e.g., roughness) can reduce the stream velocity resulting in flood reduction downstream (c.f., Hupp and Osterkamp, 1996). However, some studies suggest that revegetating floodplains, by reducing local velocities, can also lead to rise in water depth upstream (Wang and Wang, 2007). In addition to reducing floods, floodplain vegetation can minimize sediment and nutrient loading by taking up and processing nutrients and trapping sediments (Dosskey et al., 2010). Through its effect on velocity of overbank flows, floodplain re-vegetation also has the ability to reduce stream power (i.e., the rate of energy expenditure along stream; Bagnold, 1966) locally and in downstream reaches. Reduction of stream power in downstream reaches may minimize stream incision and bank collapse (Beschta and Platts, 1986). This in-turn may lead to decrease in delivery of sediment bound nutrients (e.g., phosphorus) downstream (Sekely et al., 2002). Recently, Dixon et al. (2016), using a heuristic hydrological modeling approach at a large spatial scale (>10 km^2), showed that landscape-based reforestation can significantly reduce flood peaks at a watershed scale. However, our understanding of how floodplain re-vegetation may influence flood depth and stream power has been limited to idiosyncrasies of a few reaches, and it remains unclear how the influence of revegetation on hydro-geomorphic response of streams may vary along multiple reaches at large spatial scales.

Increasing stream-floodplain connectivity is another critical stream intervention. In this context, connectivity refers to the exchange of water, nutrients, organic matter and biota between streams and floodplains (Opperman et al., 2010). Stream-floodplain connectivity can be increased by removing berms and dykes (Gergel et al., 2002), or by lowering the floodplain (Baptist et al., 2004). In general, greater accessibility of floodplains to streams during flooding can result in dissipation of energy, reduction in velocity, and changes in water depth locally and downstream (Rijke et al., 2012; Jacobson et al., 2015). Further, greater connectivity can also provide more opportunities for settling of sediments and particulate bound nutrients (e.g., phosphorous) on floodplains (Noe and Hupp, 2005). A number of studies have simulated the influence of floodplain reconnection on flood peak attenuation (Woltemade and Potter, 1994; c.f., Sholtes and Doyle, 2011), and have collectively emphasized the sensitivity of stream-floodplain properties (e.g., width, slope, and length) on flood peaks. Many of these previous studies however, either involve watershed-wide interventions or a narrow focus on responses along a few reaches. Thus, there remains a need for more nuanced understanding of how changes in stream connectivity can influence hydro-geomorphic responses along several reaches at a large spatial scale.

The effectiveness of stream interventions in attaining a desired outcome (e.g., reduction in flooding) depends on understanding the underlying processes and drivers mediating hydrological, ecological and geomorphic responses of floodplains (Ward et al., 2001;Palmer et al., 2005;Beechie et al., 2010). Interventions are likely to alter the fundamental forms and functions of floodplains, so knowing the driving processes may help in sustaining those basic properties of the ecosystem. In particular, investigating the influences of stream intervention along multiple reaches can provide us opportunities to relate these stream responses to their corresponding geomorphic characteristics within a watershed. Understanding these interactions may assist practitioners and policy makers to target interventions where they are mostly likely to make a positive difference.

To address these research gaps, we used scenario-based modeling to identify potential flooding and water quality benefits of revegetating floodplains and increasing floodplain connectivity in two watersheds of Vermont. We studied a suite of stream responses, including water depth and stream power, to compare their sensitivities to interventions. This work advances our understanding of how nature-based solutions could affect ES provided by stream-floodplain systems. It develops a novel framework that integrates stream restoration with ES, and it provides a simple screening approach to guide natural resource managers in targeting interventions to maximize intended outcomes.

Specifically, two primary questions guide this study: a) How may stream responses (water depth and stream power) vary with floodplain lowering and revegetation? and b) How do geomorphic and topographic characteristics of reaches mediate these responses? We hypothesized that the revegetation intervention would lead to local increase in water depth and variable effects on stream power due to the interaction of reduction in velocity and increase in shear stress associated with a rise in water depth. We expect the connectivity intervention to lead to decline in both water depth and stream power over baseline due to greater accessibility of floodplains to stream. The combination of connectivity and revegetation scenarios may have variable effects on water depth and stream power depending upon how interventions interact and influence the stream response.

2. Methods

2.1. Study watersheds

The study was conducted in two watersheds, representative of a range of land use types and topographical characteristics, located in the Lake Champlain Basin (LCB), Vermont, US (Fig. 1; Table S1; Appendix A). Our study watersheds contribute to water quality and flooding issues of the LCB. Lewis Creek is a lowland watershed with significant agricultural land use that has high phosphorus concentrations, whereas the Mad River is a steep, mountainous forest-dominated watershed that responds rapidly to precipitation and generates high stream power and channel erosion during flood events. A large fraction of stream reaches in both study watersheds are highly incised and are laterally unstable, attributable to a long history of channel straightening and berming and the associated loss of energy-dissipating access to floodplains (Kline and Cahoon, 2010). Stream incision, such as that evident in our study watersheds, and its role in phosphorus contributions to receiving waters, has led to statewide efforts to document geomorphic conditions and promote conservation of river floodplains (Kline and Cahoon, 2010).

2.2. Hydraulic model development

We built a 1-D steady-state hydraulic model using HEC-RAS 5.03 to conduct exploratory modeling with and without interventions for our two study watersheds. HEC-RAS is a standard hydraulic model developed by US Army Corps of Engineers that can simulate hydraulic variables, such as water surface elevations and unit stream power under steady flow conditions. The 1D steady state model solves energy and flow resistance equation iteratively between cross-sections. Due to limited data requirements and rather simple modeling approach, steady-state 1D HEC-RAS models have been widely used in the scenario-based floodplain modeling work (Gergel et al., 2002; Cook and Merwade, 2009; Jacobson et al., 2015). On the other hand, unsteady state HEC-RAS modeling is quite complex and mostly used to simulate flood peaks during events, requiring flow and stage data to validate and fulfill boundary conditions (HEC-RAS, 2016). Further, unsteady models have several instability issues that makes them challenging to run and can result in large errors if not parameterized and validated properly, especially at large spatial scales (>100 km^2). We opted for a simple 1D steady state modeling approach, given the focus of this study to provide a simple screening tool to practitioners and stream restoration managers and following previously published work using a similar approach for first-order decision support (Jacobson et al., 2015).

The major inputs of the model included a high-resolution digital elevation model (i.e., LiDAR; Light Detection and Ranging), landuse based Manning's n roughness coefficients, and peak flow values at the upstream of tributaries to fulfill boundary conditions. HEC-RAS uses a network of cross-sections to extract stream geometry and topography from a LiDAR. We used a spatial intensive (~200 m apart) network of cross-sections to feed input stream geometry (e.g., width, length, depth, floodplain elevations) and roughness in the model across both study watersheds. A landuse based approach was used to derive Manning's n as recommended by (Kalyanapu et al., 2010). We used USGS empirical equations to estimate peak flow conditions that were developed for ungauged watersheds of Vermont at 5 yr and 100 yr return-intervals (Olson, 2002; Appendix A).

Each modeled cross-section represented a reach of varying length, so hereafter we refer to cross-sections as study reaches. The model included 268 reaches for Mad River and 199 reaches for Lewis Creek (Fig. 1). The model was used to conduct scenario-based simulations to study how stream response to interventions changed over baseline conditions (i.e., without interventions) for 5 yr and 100 yr return-intervals. We estimated two response variables, water depth and total unit stream power for each cross-section. Water depth was averaged over the entire cross-section. The total unit stream power (units: W/m^2) was estimated as a product of total shear stress and total average velocity over the entire cross-section (Magilligan, 1992). In other words, total unit stream power value represents sum of unit stream power over both (left and right) banks and the unit stream power of the channel. Total unit stream power (hereafter, stream power) is a hydraulic variable but exerts

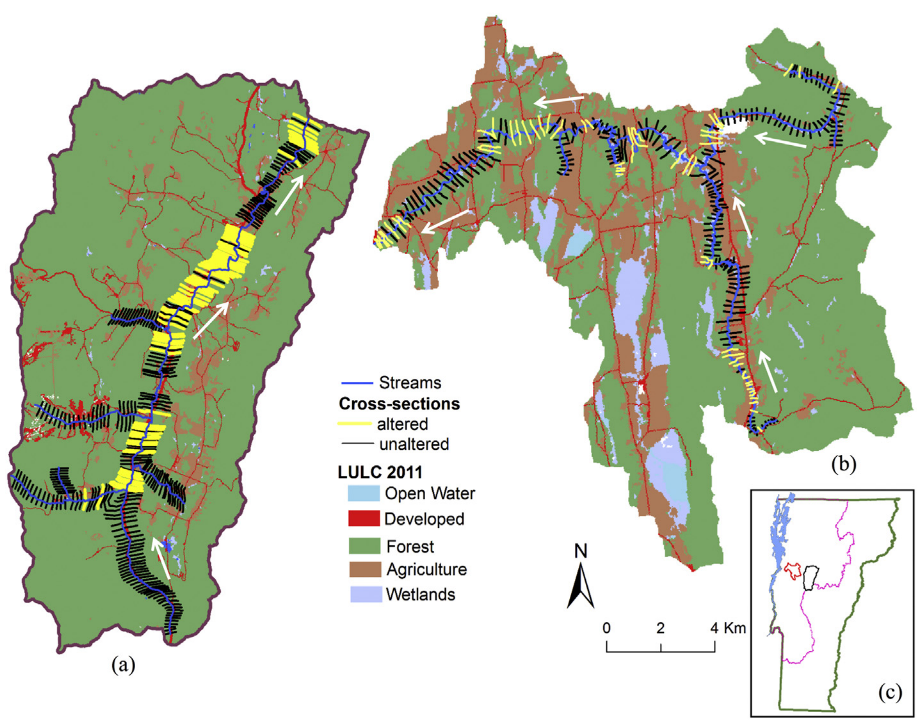

Fig. 1. Extent of HEC-RAS modeled cross-sections, altered and un-altered reaches for Mad River (a), and Lewis Creek (b) with LULC for 2011. Inset shows the state of Vermont (green solid line), the Lake Champlain (blue water body), the Lake Champlain basin boundary (pink dashed line), and the boundaries of two study watersheds (c). White arrows show the direction of flow in the watersheds.

strong control on stream morphology, and sediment transport during events (Magilligan, 1992; Thompson and Croke, 2013; Magilligan et al., 2015). Collectively, these processes may lead to channel incision resulting in impaired streams and poor water quality of the water-bodies downstream. Our stream response variables (water depth and stream power) are reasonable representations of hydro-geomorphic characteristics of streams.

2.3. Simulating floodplain interventions

Using unique statewide stream geomorphic assessment (SGA) data, we determined the morphology and physical form of reaches (Table 1) for use both in targeting modeled interventions and for subsequent statistical analysis of factors influencing intervention effectiveness. Incision ratio (IR) is a measure of vertical containment of stream and it informs us about the degree of connectivity between the stream and the adjacent floodplain (Table 1). The floodplain interventions were implemented along highly incised reaches (IR > 1.4) that did not have forested floodplains (hereafter, "altered reaches", N = 68 for Mad River; N = 57 for Lewis Creek). These altered reaches represented 25% (Mad River) and 28% (Lewis Creek) of the total modeled reaches in the study watersheds. The reaches in which no intervention was implemented are referred to as "unaltered reaches".

We simulated the influence of re-vegetating floodplains on stream response by modifying Manning's coefficient (n) along altered reaches in the hydraulic model (Fig. 2). To characterize revegetation scenarios, we used n of 0.182 representative of dense herbaceous vegetation (Kalyanapu et al., 2010). We simulated the influence of connectivity by editing the LIDAR to lower bank elevations along altered reaches (Fig. 2). The spatial extent of the lowering of floodplain was based on the combination of width of river corridor (outlined by Vermont Agency of Natural Resources) and the presence of valley walls, whereas the depth of lowering was based on water depth derived from the baseline model run at a 2 yr return period (i.e., channel forming flows, Wolman and Miller, 1960; Appendix A). Using the combination of these two interventions, we ended up with three scenarios: a) floodplain revegetation (REVEG), b) connectivity with baseline vegetation (CBVEG) scenario in which we lowered the floodplain and did not revegetate, and b) connectivity with revegetation (CRVEG) scenario in which we lowered the floodplain and re-vegetated it with n of 0.182 (Fig. 2).

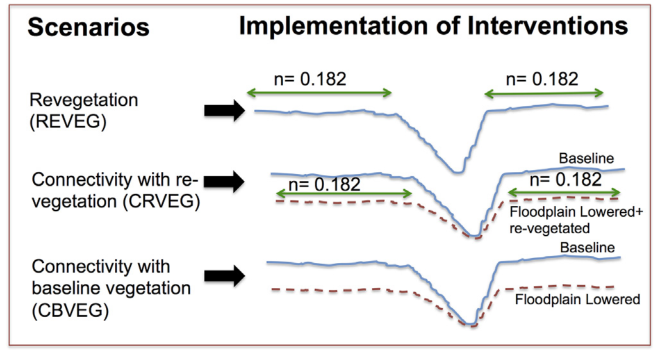

Fig. 2. Three scenarios used in the analysis. In revegetation scenario (REVEG), Manning's coefficient (n) was changed from baseline to n = 0.182. In connectivity scenarios, LIDAR was edited to lower bank elevations and no changes were made to baseline n (connectivity with baseline vegetation, CBVEG), whereas LIDAR was edited to lower bank elevations and baseline n was changed to 0.182 (connectivity with revegetation, CRVEG).

We ran the model separately for each scenario, and interventions were simulated simultaneously along all altered reaches (Fig. 1). Changes in stream response were estimated by subtracting the baseline condition from the condition due to interventions along both altered and unaltered reaches. Further, we used a statistical metric to detect a meaningful change in stream response to interventions. We computed ±2 standard error for changes in water depth and stream power over baseline for individual intervention. Further, we focused our analyses (e.g., descriptive statistics, multivariate analysis) on the stream responses that exceeded these error bounds.

To understand the influence of interventions on each stream response, we analyzed the magnitude and the direction of change in stream response from baseline, and the percentage of altered or unaltered reaches that showed increase or decrease in response over baseline. Generally, high stream power is known to accelerate channel incision leading to the impairment of streams. So, depending on the direction of change, we categorized the stream power into two groups as improved (i.e., decrease in stream power over baseline) or degraded (i.e., increase in stream power over baseline) and calculated the percentage of reaches influenced to total altered reaches or total unaltered reaches for all scenarios. Further, for each scenario, we computed the mean and standard deviation individually for altered and unaltered reaches in which stream responses over baseline exceeded the error bounds for the study watersheds.

Table 1
Summary of geomorphic and topographic variables used in the study.

Datasets	Abbreviations	Description
Geomorphic variables		
Incision Ratio	IR	Measure of vertical containment of stream, ratio of low bank height to bankfull maximum depth
Entrenchment Ratio	ER	Measure of vertical containment of stream, ratio of floodprone width to bankfull width
Stream Type	ST	Characterization of streams based on geomorphic variables e.g., slope, ER, Sinuosity, landforms, bed features. We represented them inform of numbers in the statistical analysis: A (1), B(2), C(3), D (4), E(5) and F(6)
Width to depth ratio	WDR	Measure of stream adjustment and energy available to mobilize sediments
Sinuosity	SY	Measure of movement (bending, curving) of streams along floodplain
Dominant landuse (Left and Right banks)	DLU_L, DLU_R	Landuse with highest area coverage
Sub dominant landuse (Left and Right banks)	SDLU_L, SDLU_R	Landuse with second highest area coverage
Categorical variable		
	RTYP	Response type (altered=1; unaltered =0)
Topography variables		
Elevation	ELV	Elevation at cross-section
Slope	SLP	Slope at cross-section
Drainage area	DA	Contributing area for each cross-section
Bed Gradient	BG	Ratio of difference in elevation between two adjacent cross-sections to distance between both cross-sections

2.4. Geospatial, statistical and sensitivity analyses

The stream geomorphic assessment (VANR, 2009) datasets include some key variables that can be helpful in understanding the stream response to interventions such as, entrenchment ratio, width to depth ratio, and stream type (Table 1) For instance, entrenchment ratio provides an index of available floodplain width normalized by the channel width, with higher entrenchment ratios corresponding to less confined channels. In addition to the geomorphic variables (Table 1), we also estimated elevation, drainage area, slope and bed gradient for all study reaches using DEM (10 m x10m). We also used a categorical variable "response type" (altered =1, unaltered =0) as a factor to test whether the difference noted in stream response between altered and unaltered were substantial enough to explain the spatial patterns of change in water depth and steam power over baseline.

We conducted Redundancy analysis (i.e., multivariate-ordination analysis), as recommended by (Bhattacharya and Osburn, 2017; Appendix A), to understand how the geomorphic and topographic variables (Table 1) related to the magnitudes and directions of stream response for altered and unaltered reaches during all scenarios. We summarized Redundancy analysis results in the form of bi-plots and a summary table showing two major components that explained most of the variance in responses and the key explanatory variable derived from the analysis. Redundancy analysis was conducted with "Vegan 2.4–4" (Oksanen et al., 2018) in R statistical software (R Core Team, 2013).

We tested the sensitivity of the modeled responses to roughness (i.e., Manning's n) and flow conditions used for flood recurrence-intervals used in the model across both study watersheds. We altered the Manning's n by ±10% along all modeled reaches for 100 yr and 5 yr flood recurrence intervals and estimated the percentage change in water depth and stream power from the baseline model. Similarly, we varied the flows by ±10% for 100 yr and 5 yr flood recurrence-intervals and estimated the percentage change in water depth and stream power from the baseline model.

3. Results

3.1. Stream response to interventions along altered reaches

Our findings showed that interventions influenced stream responses in diverse ways depending upon the individual response, scenarios and flood return-intervals (Figs. 3, 4, A.1, A.2; Table 2). On average, the revegetation (REVEG) scenario exhibited an increase in water depth over baseline conditions along ~80% of altered reaches in both watersheds (Table 2; Figs. 3–5) and <6% showed a decrease in water depth over baseline. Similarly, the connectivity with revegetation (CRVEG) scenario resulted in increases in water depth over baseline along 90% (Mad River) and 60% (Lewis Creek) of altered reaches, whereas 5–20% of altered reached showed decline in water depth from baseline. The connectivity with baseline vegetation (CBVEG) scenario resulted in decrease in water depth over baseline along 46% (Mad River) and 43% (Lewis Creek) of altered reaches, whereas <10% of altered reaches showed an increase in water depth over baseline condition (Table 2; Figs. 3–5). The number of reaches that showed change in water depth over baseline during both connectivity scenarios differed by flood recurrence-intervals (Fig. 5). The magnitude of change in water depth was always greater for 100 yr than 5 yr flood return-intervals and REVEG

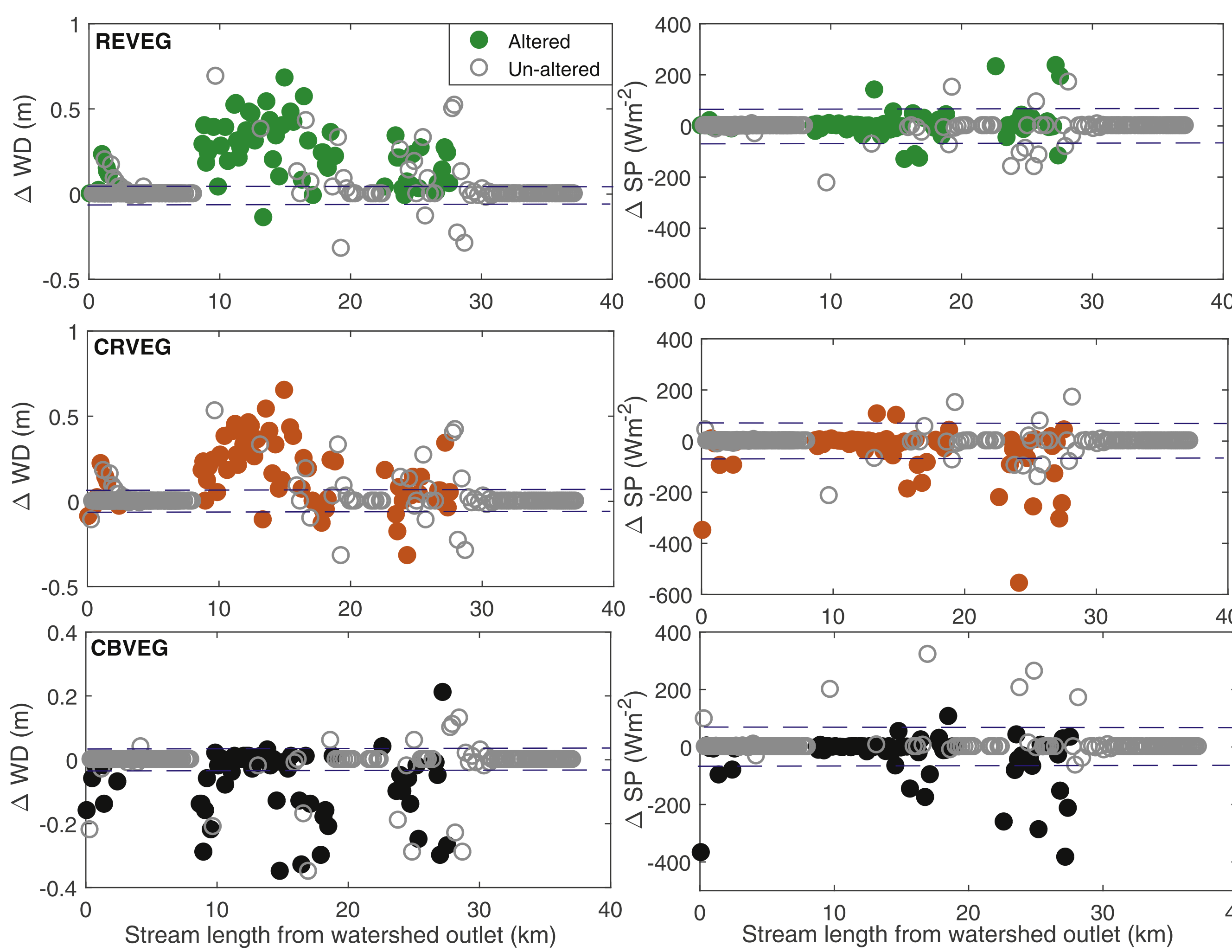

Fig. 3. Changes in water depth (ΔWD) and stream power (ΔSP) over baseline conditions in the Mad River. Data are shown for altered (filled circles) and unaltered (grey open circles) reaches for 100 yr flood return-interval. Blue dashed lines represent ±2 standard error for each modeled response. Responses above and below these error bounds were considered as meaningful change from the baseline. REVEG: Revegetation, CRVEG: Connectivity with revegetation, CBVEG: connectivity with baseline vegetation.

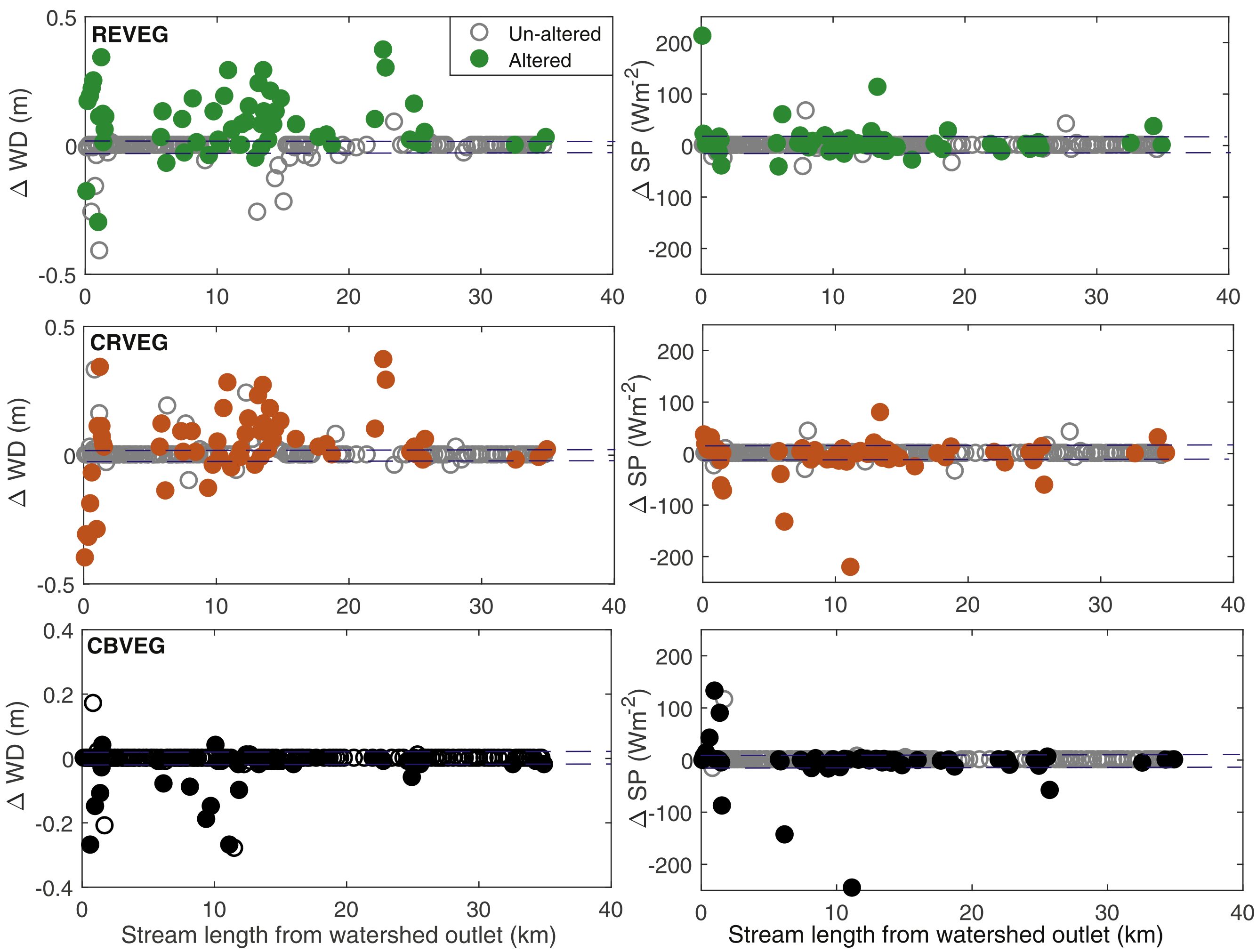

Fig. 4. Changes in water depth (ΔWD) and stream power (ΔSP) over baseline conditions in the Lewis Creek. Data are shown for altered (filled circles) and unaltered (grey open circles) reaches for 100 yr flood return-interval. Blue dashed lines represent ±2 standard error for each modeled response. Responses above and below these error bounds were considered as meaningful change from the baseline. REVEG: Revegetation, CRVEG: Connectivity with revegetation, CBVEG: connectivity with baseline vegetation.

scenario showed highest absolute change in water depth in both watersheds (Table 2).

Overall, the direction of change noted in stream power over baseline due to intervention was mostly consistent among scenarios (Figs. 3, 4; Table 2), but the number of reaches that showed changes was highly variable among scenarios and between watersheds (Fig. 5). In general, all scenarios showed decrease in stream power along most of reaches over baseline and the magnitude of change was greater for both connectivity scenarios than the revegetation (REVEG) scenario in both watersheds (Table 2). In particular, CRVEG scenario showed a decline in stream power over baseline (i.e., improved) along ~34% of altered reaches and an increase in stream power over baseline (i.e., degraded) along 14% of altered reaches (Fig. 5). The revegetation scenario (REVEG) showed decline in stream power over baseline (i.e., improved) along 24% of altered reaches and increase in stream power over baseline (i.e., degraded) along 21% of altered reaches. There was no consistent pattern in improved and degraded reaches with flood recurrence intervals among scenarios in the study watersheds, but the magnitude of

Table 2
Mean (Std. deviation) of change in water depth (ΔWD) and stream power (ΔSP) from baseline due to interventions along altered reaches for the study watersheds.

Mad river	5 yr			100 yr		
Mean (Std. dev)	REVEG	CRVEG	CBVEG	REVEG	CRVEG	CBVEG
ΔWD (m)	0.14(0.1)	0.12(0.11)	−0.04(0.07)	0.24(0.16)	0.17(0.11)	−0.15(0.17)
ΔSP (Wm^{-2})	−0.08(54)	−31(83)	−30(59)	−48(104)	−162(159)	−184(161)
Lewis creek	5 yr			100 yr		
Mean (Std. dev)	REVEG	CRVEG	CBVEG	REVEG	CRVEG	CBVEG
ΔWD (m)	0.08 (0.07)	0.04(0.09)	−0.05(0.11)	0.1(0.12)	0.03(0.15)	−0.16(0.23)
ΔSP (Wm^{-2})	−7(51)	−41(89)	−54(110)	29(94)	−1(88)	−17(72)

REVEG: Revegetation; CRVEG: Connectivity with Revegetation, CBVEG: Connectivity with baseline vegetation.

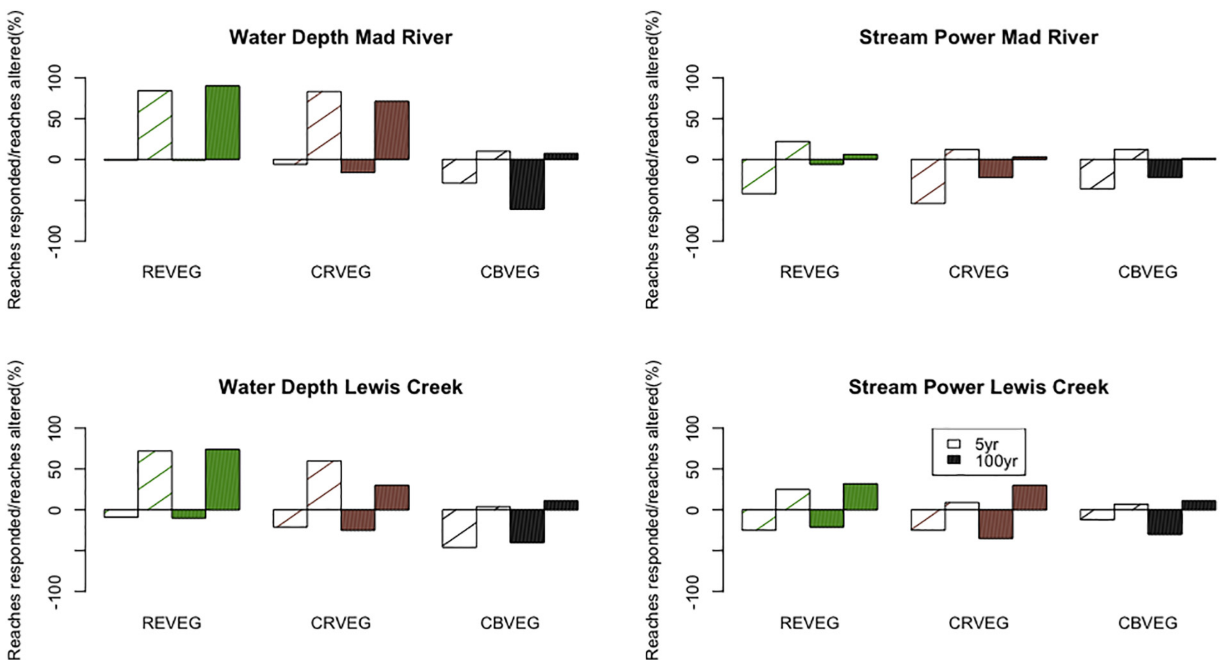

Fig. 5. The proportion (%) of altered reaches that showed increases (+ve%) or deceases (−ve%) in water depth and stream power over baseline across interventions during 5 yr and 100 yr return-intervals for the study watersheds. Low density and high density of hatched lines represent 5 yr and100 yr return-intervals. REVEG: Revegetation, CRVEG: Connectivity with revegetation, CBVEG: connectivity with baseline vegetation.

change increased by >200% (Mad River) and decreased by >300% (Lewis Creek) from 5 yr to 100 yr flood return-intervals (Table 2).

3.2. Stream response to interventions along un-altered reaches

Overall, scenarios influenced 1–16% of total unaltered reaches (Mad River: 201; Lewis Creek: 142) in which no interventions were made in the study watersheds (Fig. 6; Table A.2). Most of these unaltered reaches were immediately upstream or downstream of the altered reaches (Figs. 3, 4). The direction of change noted in stream response over baseline was similar to altered reaches for both revegetation based interventions. For instance, REVEG and CRVEG scenarios lead to increase in water depth along 9–14% of unaltered reaches and a decrease in water depth along 2–4% of unaltered reaches in both watersheds. CBVEG resulted in decrease in water depth along 4% and 2% of unaltered reaches and increase in water depth only along 1–2% of unaltered reaches. Changes in water depth were noted up-to 500 m upstream of the altered reaches and resulted in backwater effect (i.e., rise in water depth upstream) along unaltered reaches (Fig. A.3). The backwater effect was more pronounced in REVEG scenario and minimal in CBVEG scenario. A number of unaltered reaches influenced were substantially higher for water depth during 100 yr and 5 yr flood return-intervals across both watersheds (Fig. 6).

In general, the changes noted in stream power from baseline were not consistent among scenarios, between watersheds and flood return-intervals. REVEG and CRVEG scenarios lead to a decrease in stream power (i.e., improved) along 4–7% of total unaltered reaches and an increase in stream power over baseline (i.e., degraded) along 1–3% of unaltered reaches in both watersheds (Fig. 6; Table A.2). CBVEG showed an increase along 1–3% and a decrease along 1% of total unaltered reaches. In other words, both revegetation scenarios resulted in substantial number of improved reaches from baseline but not the CBVEG scenario.

3.3. Geomorphic and topographic influences on stream response

Redundancy analysis showed how geomorphic variables influenced stream response to scenarios along altered and unaltered reaches (Fig. 7, Table 3). Total explanatory power of change in stream response to interventions varied from 22% to 56% where significant ($p < 0.05$) explanatory power was noted more frequently for water depth than stream power during most of the scenarios (Table 3). There were several geomorphic variables that explained change in water depth and stream power over baseline but some of the variables consistently showed significant influence on stream responses (Table 3). These variables include: entrenchment ratio, sinuosity width to depth ratio, bed elevation, slope, and drainage area of reaches.

For revegetation scenario (REVEG) in Mad River, RDA component 1 explained 29% of the spatial patterns of water depths and some of the variance could be attributed to entrenchment ratio and drainage area (Fig. 7a). Further, 45% of variance in stream power was explained by RDA component 1 attributed mostly to response type (a categorical variable) and incision ratio (Fig. 7b, Table 3). Similarly, for REVEG scenario in Lewis Creek, RDA component 1 explained 25% of the spatial patterns of water depths and some of the variance could be attributed to sinuosity (Fig. 7c). For connectivity with revegetation scenario in Lewis Creek, RDA component 1 explained 18% (water depth) and 24% (stream power) and most of their variances could be attributed to elevation and slope of reaches (Table 3).

3.4. Sensitivity analysis

In general, our findings indicated that the sensitivity of the modeled responses to model parameters (discharge and Manning's n) varied widely for both watersheds (Table A.3). Water depth was more sensitive to discharge with median change of ±5% from the baseline model than Manning's n with a median change of ±4%. Similarly stream power was more sensitive to discharge with median change ±10% from the baseline model in comparison to ±4% for Manning's n. The range of these median values was higher for Mad River than Lewis creek. Our analysis focused mostly on understanding the direction and magnitude of change in stream response from baseline and not on forecasting the absolute magnitude of flood or stream power. The small range of uncertainty noted here may not have significant influence on our scenario based modeling work.

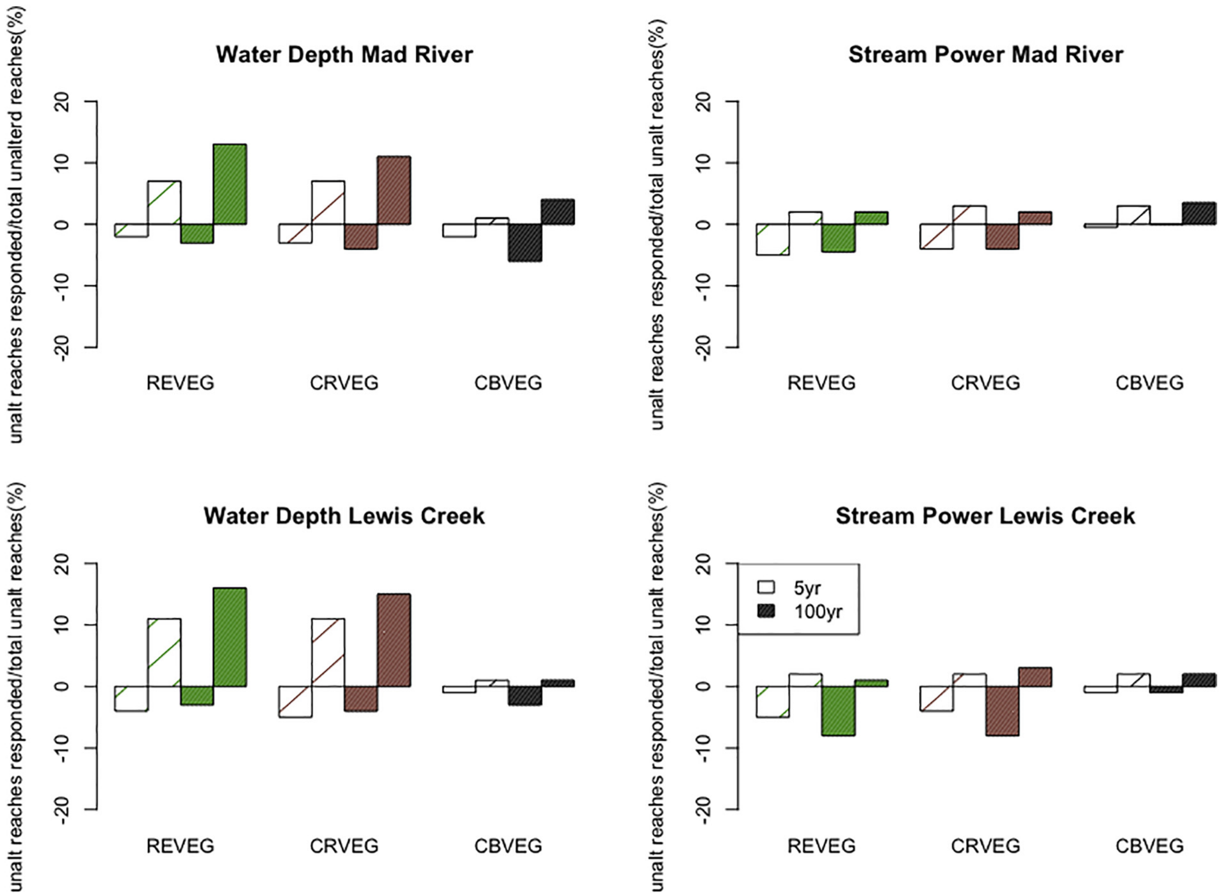

Fig. 6. The proportion (%) of unaltered reaches (Mad River: 201; Lewis Creek: 142) that showed increases (+ve%) or decreases (−ve%) in water depth and stream power over baseline across interventions in Mad River and Lewis Creek during 5 yr and 100 yr return-intervals. Low density and high density of hatched lines represent 5 yr and 100 yr return-intervals. REVEG: Revegetation, CRVEG: Connectivity with revegetation, CBVEG: connectivity with baseline vegetation.

4. Discussion

Our work is predicated on an interest in evaluating how restoration practices that attempt to recover natural functions of river corridors can restore the ecosystem services provided by these ecosystems with associated benefits for flood mitigation and nutrient retention. “Natural” vegetation (e.g., forests) along riparian corridor provides a service of slowing runoff, allowing infiltration and nutrient update. Connected floodplains allow dissipation of energy during flood flows, allowing water to infiltrate and particulate-bound and dissolved nutrients to be trapped on floodplains. The simple screening approach we developed here allows us to look at the effects of revegetation and floodplain reconnection on two metrics, water depth and stream power, that index these important ecosystem services. Our findings indicated that most of the scenarios led to decrease in stream power and increase in water depth and both can improve water quality by reducing erosion and extending time for particulates to settle on floodplains. These changes, however, may come with subsequent flooding locally and upstream of the study reaches (Figs. 3, 4, A.3). Further, floodplain-based stream interventions can result in either improvement or degradation of reaches from baseline (Figs. 5, 6), depending upon how reach-scale geomorphic and topographic characteristics interact with interventions (Fig. 7; Table 3).

4.1. Stream response to interventions along altered and unaltered reaches

Overall, increased hydraulic friction due to the re-vegetation scenario can lead to reduction in stream velocity and increase in residence time and subsequent rise in water depth (Figs. 3, 4). This phenomenon explained the backwater effect that was more frequently noted along unaltered reaches situated upstream of the altered reaches in re-vegetation scenario (Fig. A.3) as noted in other studies. Wang and Wang (2007) measured decline in stream velocity and rise in water depth upstream of study reaches due to floodplain vegetation in China. Thomas and Nisbet (2007) simulated the influence of floodplain vegetation and reported rise in water depth up-to 400-m upstream and decline in stream velocity along a 2 km reach in UK. The revegetation scenario (REVEG) showed a decrease in stream power (Table 2). Unit stream power is a product of velocity and shear stress along cross-section, so a decline in stream velocity due to greater friction can likely explain the decrease in stream power.

The lowering of floodplains led to the reconnection of streams with adjacent banks and provided more access to floodplains, resulting in decrease of stream responses (Figs. 3, 4) and minimal backwater effect during connectivity scenarios compared to the revegetation scenario (Fig. A.3). Baptist et al. (2004) showed that together lowering of floodplain and revegetating floodplains in succession with different types of vegetation could decrease flood inundation area and sedimentation for a reach of Rhine River in Netherlands.

4.2. Influence of geomorphic and topographic variables on stream responses

Similar interventions were applied along many reaches, but the magnitude and direction of stream response varied widely among reaches within the study watersheds (Figs. 4, 5), highlighting the controls of reach scale morphology and topographic settings of watersheds on interventions (Fig. 7; Table 3). Among geomorphic variables we

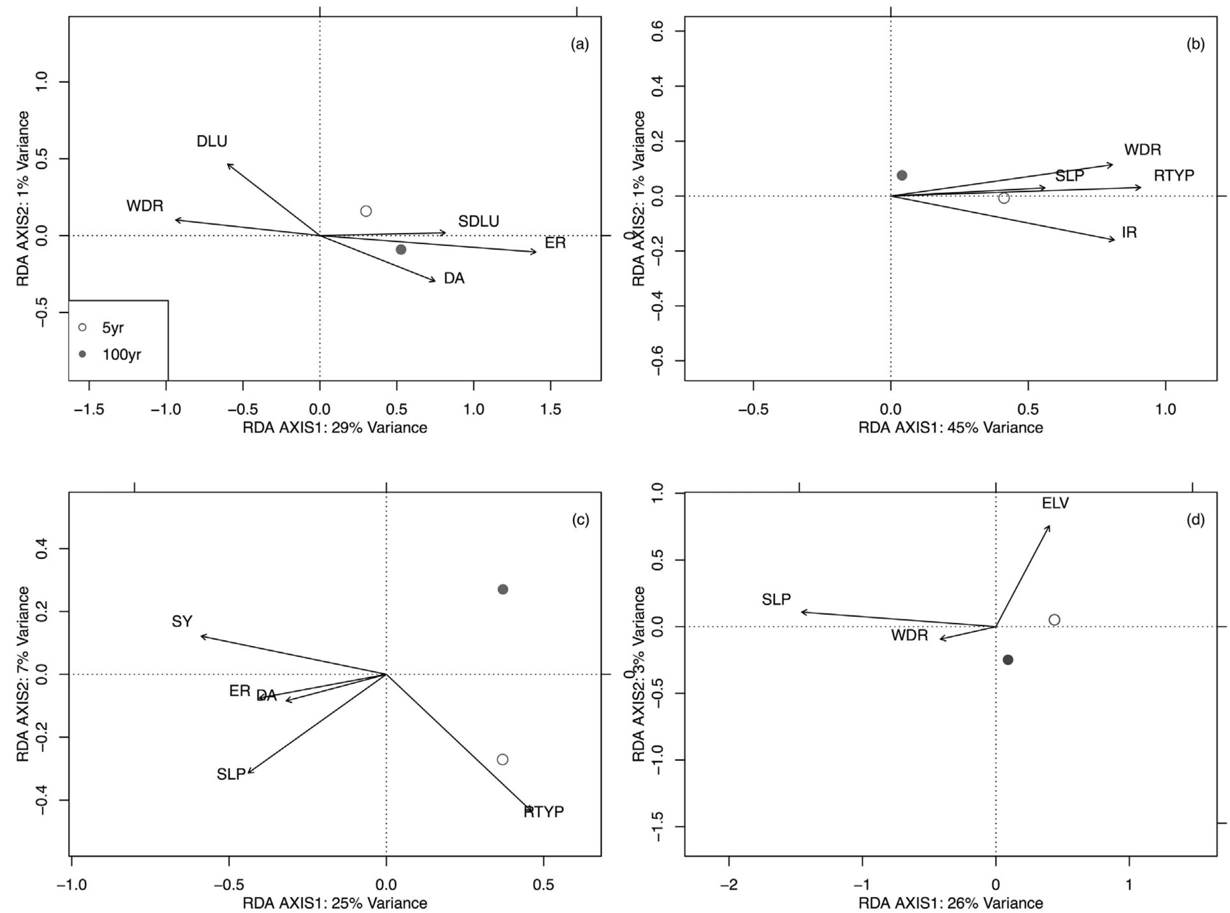

Fig. 7. Bi-plots relating changes in stream response to geomorphic and topographic variables. Each panel shows results of a RDA analysis testing the effects of geomorphic and topographic variables on the pattern of predicted changes in stream responses over baseline. Top row: geomorphic and topographic influences on changes in water depth a), and stream power b), due to revegetation scenario in the Mad River. Bottom row: geomorphic and topographic influences on changes in water depth due to revegetation scenario in Lewis Creek c), and the influences of geomorphic and topographic variables on changes in stream power due to connectivity with revegetation scenario in Lewis Creek d). All models were significant ($p < 0.05$). Open and filled circle represent 5 yr and 100 yr flood return-intervals. Please refer to Table 1 for the abbreviations.

examined (Table 1), we found that entrenchment ratio, sinuosity, and width to depth ratio were important controls on the direction and magnitude of water depth due to interventions (Fig. 7; Table 3). Entrenchment ratio is a measure of confinement for a reach (Rosgen, 1994), and bigger the entrenchment ratio the less confined the reach and greater accessibility to the adjacent floodplain (Table 1). This can lead to greater change in water depth in response to intervention. Sinuosity restricts the conveyance of flow, leading to rise in water depth locally and upstream (Hu et al., 2017). So, the implementation of interventions may not show substantial change in water depth from already high depth in the baseline condition. The greater changes in water depth in the middle reaches of the Mad River could be attributed to their relatively small width to depth ratio (~20), where most of the interventions resulted in remarkably higher change in stream response from baseline (Figs. 3, 4). Overall, these results suggest exercising caution when implementing interventions and indicate that greater benefits in terms of flood reduction can be attained while intervening along relatively less confined and low sinuous reaches.

Table 3
Summary of all Redundancy analysis for changes in water depth (ΔWD) and stream power (ΔSP) from baseline due to all interventions.

Watershed	Scenario	Stream Response	Variance Explained Total (Components)*	Significant Model Variables	Variable with highest R^2
Mad River	REVEG	ΔWD	30% (C1:29%, C2:1%)	ER, DA, WDR, DomLULC	ER(20%)
		ΔSP	46% (C1:45%, C2:1%)	IR, WDR, SLP, RTYP	IR(15%)
	CBVEG	ΔWD	NS		
		ΔSP	56% (C1:55%; C:1%)	DA, SLP, RTYP	DA(17%)
		ΔWD	29% (C1:28%; C:1%)	ER, WDR	ER(18%)
	CRVEG	ΔSP	NS		
Lewis Creek	REVEG	ΔWD	32% (C1:25%, C2: 7%)	DA, SLP, SY, ER, RTYP	SY (14%)
		ΔSP	NS		
	CBVEG	ΔWD	40% (C1:40%)	ELV	ELV
		ΔSP	NS		
	CRVEG	ΔWD	22% (C1:18%; C2:4%)	ELV, SLP, ER, BG	ELV (12%)
		ΔSP	26% (C1:24%, C2: 2%)	ELV, SLP	SLP (24%)

* C1 and C2 are two major components (RDA AXES) of the models; All models are significant $p<0.05$, except Mad River stream power model for REVEG scenario with $P=0.06$. NS is non-significant model $P>0.2$. Please refer to Table 1 for abbreviations.

Elevation, drainage area, and slope of reaches showed strong effects on the changes in stream power due to interventions, highlighting that intervention effectiveness may differ with topographic settings (Table 3; Fig. 7). Generally, stream power is a function of slope and drainage area (Knighton, 1998), so the strong influence of these variables for changes in stream power due to intervention are highly likely. High elevation streams have relatively steep bed gradient, such as in Mad River, that can result in sudden changes in energy gradient influencing stream power (Wohl et al., 2004). These results indicate that restoring high elevation headwater reaches may provide greater benefits in terms of reducing stream power, resulting in improving water quality in downstream water-bodies.

Overall, our findings suggested that geomorphic and topographic characteristics of reaches influenced stream response and could determine the effectiveness of interventions. The stream restoration community has argued for developing process based interdisciplinary understanding before implementing intervention (Kondolf et al., 2003; Beechie et al., 2010; Palmer et al., 2010). Our work supports this argument and suggests that the more we understand the major processes and drivers of stream ecology, morphology and hydrology, the more likely restoration interventions will succeed.

4.3. Effectiveness and tradeoffs of interventions

Our findings showed that the effectiveness of interventions depends upon the criteria used to evaluate interventions (Fig. 5). Overall, water depth increased and decreased over baseline along 47% and 28% of altered reaches, respectively, whereas stream power increased and decreased from baseline along 14% and 28% of altered reaches, respectively. These findings suggested that the effectiveness of interventions can vary significantly between stream responses (Fig. 5) and further highlighted the need for pre- and post-monitoring, as stream interventions may not necessarily lead to successful outcomes (Palmer et al., 2005).

The difference in stream response to connectivity scenarios highlighted the importance of vegetation type chosen for revegetating floodplains (Figs. 3, 4, A.3). Mostly, because the vegetation type can exacerbate local flooding and reduce potential flood mitigation benefits of re-connecting floodplains. Consistent increase in water depth over longer duration, locally and upstream of the altered reaches, can influence the biodiversity of floodplain ecosystems (Wootton, 2012; c.f., Rolls et al., 2017). Further, the slow conveyance of flow along reach due to vegetation during events can have significant impact on the stream morphology (van Dijk et al., 2013; Hu et al., 2017). On the positive side, the greater residence time and the standing water column on floodplain provides further opportunities for sediments and nutrients to settle or to be trapped by vegetation (Dekker et al., 2016).

In terms of stream power, both connectivity scenarios resulted in greater decline in stream power and could be useful in addressing impaired and degraded reaches. It is worth recognizing that connectivity scenarios led to decline in stream power up-to 600 W/m^2. Previous study shows that scouring of channel may begin at 300 W/m^2 (Baker and Costa, 1987) and the stream power >300 W/m^2 could be detrimental to the stream morphology during extreme events (Magilligan, 1992). Our work highlights the critical role that these interventions may play in mitigating stream incision and undercutting during events.

The effectiveness of intervention could also be evaluated on the basis of how many unaltered reaches exhibited changes in stream response due to interventions (Fig. 6). The changes noted in stream response, along upstream or downstream of altered reaches, are more likely to occur on privately owned lands. Hence, the response of interventions on these non-conserved reaches needs to be carefully studied. To, summarize, each intervention has both positive and negative influences on floodplains, so we need to evaluate their tradeoffs on stream response and then select the intervention that addresses our ecological restoration and management needs. This approach may result in the least possible unintended consequences on overall health of the riparian corridor.

Several limitations of our work deserve mention. The representation of vegetation friction in a single number (e.g., Manning's n) is a very simplistic view of a much complex process and may not account for all possible ways in which vegetation biomechanics may influence stream response to revegetating floodplains. We need more work to develop accurate representation of these factors in hydraulic models at large spatial scale and it remains one of the critical research needs of geomorphology (Curran and Hession, 2013). Use of 2D hydraulic models might have provided more useful insights than 1D, but the implementation of 2D model would require high spatial resolution stream bathymetry that are not available in Vermont. Despite these limitations, the insights gained here could be useful for practitioners and policymakers alike to gauge the complexity of restoration processes and exercise caution when implementing such interventions.

4.4. Management implications

The effectiveness of interventions varied with individual stream responses suggesting that each intervention may influence various aspects of hydro-geomorphic responses differently, further underscoring the need of monitoring multiple abiotic and biotic responses to gain comprehensive understanding of stream response to interventions (Wootton, 2012). These results can also inform several river management practices that are widely used worldwide. For example, river managers often implement stream interventions with an aim to address a geomorphic issue for a small reach, but the targeted interventions are likely to influence abiotic aspects of the stream ecosystem upstream and downstream of the altered reach. Thus, such management practices should be conducted with caution.

Revegetating floodplains remains a controversial intervention as it can lead to flooding and backwater effects (Thomas and Nisbet, 2007), which our models also indicate. Our findings suggest that interventions could be targeted on reaches where upstream reaches are not incised and confined, to minimize upstream and local flooding. This approach may also help to minimize negative influences on the local biodiversity that may not survive frequent episodic inundations over long term. Such strategically selective approaches for stream restoration are supported with previous work (Rohde et al., 2006; Leyer et al., 2012). Thus, spatially explicit evaluation of targeted reaches in response to interventions can lead to more effective and efficient implementation of restoration activities.

Given the spatial heterogeneity in stream response to interventions (Figs. 3, 4), we may not be able to generalize ecosystem services provided by these reaches based on their morphology such as stream type. Generalization of services is further complicated because different stream reaches are important for different services. For instance, intervention along a headwater reach may be more beneficial in minimizing stream power but may not provide flooding benefits. Further, the absence of stream class as a predictor of stream responses suggests that implementing interventions solely on a stream classification approach may not lead to desired outcome. Thus, our results do not encourage implementing stream interventions based on a stream classification approach. Lastly, this work was conducted in a close collaboration with practitioners and conservationists to encourage much needed multi-stakeholder stream restoration efforts (Bernhardt et al., 2007). This work may greatly benefit the stream restoration and management community worldwide.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2018.03.198.

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